Woody debris and tree regeneration dynamics following severe wildfires in Arizona ponderosa pine forests

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Abstract: Severe forest fires worldwide leave behind large quantities of dead woody debris and regenerating trees that can affect future ecosystem trajectories. We studied a chronosequence of severe fires in Arizona, USA, spanning 1 to 18 years after burning to investigate postfire woody debris and regeneration dynamics. Snag densities varied over time, with predominantly recent snags in recent fires and broken or fallen snags in older fires. Coarse woody debris peaked at > 60 Mg/ha in the time period 6–12 years after fire, a value higher than previously reported in postfire fuel assessments in this region. However, debris loadings on fires older than 12 years were within the range of recommended management values (11.2–44.8 Mg/ha). Overstory and regeneration were most commonly dominated by sprouting deciduous species. Ponderosa pine (*Pinus ponderosa* C. Lawson var. *scopulorum* Engelm.) overstory and regeneration were completely lacking in 50% and 57% of the sites, respectively, indicating that many sites were likely to experience extended periods as shrublands or grasslands rather than returning rapidly to pine forest. More time is needed to see whether these patterns will remain stable, but there are substantial obstacles to pine forest recovery: competition with sprouting species and (or) grasses, lack of seed sources, and the forecast of warmer, drier climatic conditions for coming decades.

Résumé : Partout dans le monde les feux de forêt sévères laissent derrière de grandes quantités de bois mort et de régénération qui peuvent influencer la trajectoire future des écosystèmes. Nous avons étudié une chronoséquence de feux sévères en Arizona, aux États-Unis, allant de 1 à 18 ans après un feu pour examiner la dynamique de la régénération et des débris ligneux après feu. La densité des chicots variait dans le temps : les chicots récents étaient surtout associés aux feux récents et les chicots cassés ou renversés étaient surtout associés aux feux plus vieux. Les débris ligneux grossiers ont culminé à plus de 60 Mg/ha durant la période de 6–12 ans après feu; c'est une valeur plus élevée que celle qui a été rapportée dans les évaluations des combustibles après feu dans cette région. Cependant, les charges de débris associés aux feux plus vieux que 12 ans se situaient dans les limites des valeurs recommandées en aménagement (11,2–44,8 Mg/ha). L'étage dominant et la régénération étaient le plus souvent dominés par des espèces décidues qui produisent des rejets. L'étage dominant et la régénération de pin ponderosa (*Pinus ponderosa* C. Lawson var. *scopulorum* Engelm.) étaient complètement absents dans respectivement 50 et 57 % des stations, indiquant que plusieurs stations allaient probablement passer de longues périodes sous forme d'arbustaie ou de prairie plutôt que de retourner rapidement à une forêt de pin. Il faut plus de temps pour savoir si ces patrons demeureront stables mais il y d'importants obstacles au retour de la forêt de pin : la compétition des espèces qui produisent des rejets ou celle des graminées, l'absence de sources de graines, ainsi que les conditions climatiques plus chaudes et plus sèches prévues au cours des prochaines décennies.

[Traduit par la Rédaction]

Introduction

Wildfire events initiate complex sequences of dead wood decay and development of new fuel and habitat structures over time (Haslem et al. 2011). Contemporary fires in southwestern US forests are increasingly large and severe (Littell et al. 2009; Miller et al. 2009), paralleling trends in Europe and Asia (Pausas et al. 2008). The postfire environment is characterized by woody debris and regenerating plant communities, the structure and composition of which contribute to future fire behavior and effects (Passovoy and Fulé 2006). These attributes also comprise important habitat elements such as snags (Chambers and Mast 2005), substrate for wood-decay fungi (Cornwell et al. 2009), and plant communities (Swanson et al. 2010). In contrast to ecosystems adapted to infrequent, severe fire disturbance such as Chilean *Nothofagus* forests (Veblen et al. 2008), southwestern US forests are dominated by species adapted to frequent, low-severity fires (Barton 2002). The fuel complex of woody debris and postfire regeneration left behind following uncharacteristi-

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Table 1. Species names and codes used in the text.

Species	Common name	Code
Abies concolor (Gord. & Glend.) Lindl. Ex Hildebr.	White fir	ABCO
Juniperus deppeana Steud.	Alligator juniper	JUDE
Juniperus osteosperma (Torr.) Little	Utah juniper	JUOS
Pinus edulis Engelm.	Twoneedle pinyon	PIED
Picea engelmannii Parry ex Engelm.	Engelmann spruce	PIEN
Pinus ponderosa C. Lawson var. *scopulorum* Engelm.	Ponderosa pine	PIPO
Populus tremuloides Michx.	Quaking aspen	POTR
Pseudotsuga menziesii (Mirb.) Franco	Douglas-fir	PSME
Quercus arizonica Sarg.	Arizona white oak	QUAR
Quercus chrysolepis Liebm.	Canyon live oak	QUCH
Quercus gambelii Nutt.	Gambel oak	QUGA
Quercus grisea Liebm.	Gray oak	QUGR
Quercus hypoleucoides A. Camus	Silverleaf oak	QUHY
Quercus rugosa Née	Netleaf oak	QURU
Robinia neomexicana A. Gray	New Mexico locust	RONE

Note: Nomenclature follows USDA plants database (USDA 2011).

cally severe wildfires in southwestern US forests poses important ecological and management issues. Retaining woody debris has the advantages of minimizing disturbance to the recently burned site, maintaining high levels of organic material for wildlife habitat and soil development, and avoiding opposition to postfire logging (McIver and Starr 2001; Beschta et al. 2004). Postwildfire coarse woody debris (CWD) is often considered to represent a hazardous level of fuel that could support reburning (e.g., U.S. Department of Agriculture (USDA) 2004), although Thompson and Spies (2009) found that a salvage-logged and replanted forest burned more severely than the surrounding forest. Removal of fire-killed timber may also bring economic benefits (McIver and Starr 2001).

Despite the importance of these issues, underscored in 2011 by the largest fires ever recorded in Arizona (217 750 ha) and New Mexico (63 400 ha), relatively little quantitative information exists about the quantity and quality of postwildfire fuels in southwestern forests. With limited empirical data, managers are often left with rough estimates (USDA 2004) or anecdotal accounts as the basis for making complex and controversial decisions about postfire management such as deciding to remove fuels through logging or to retain fuels on site. Information regarding the amount, arrangement, potential fire hazard, and dynamics of postwildfire fuels is limited to a few studies of tree mortality and (or) snag dynamics (e.g., Harrington 1996; Chambers and Mast 2005; Ganey and Vojta 2005, 2011). Only one study has specifically looked at unmanaged (unlogged) postwildfire fuels in the US Southwest, around the San Francisco Peaks in northern Arizona (Passovoy and Fulé 2006). On a chronosequence of seven fires in forests previously dominated by ponderosa pine (*Pinus ponderosa* C. Lawson var. *scopulorum* Engelm.) (see Table 1 for scientific names of species), CWD ranged from 3.3 Mg/ha to a high of 41.3 Mg/ha (Passovoy and Fulé 2006); these values did not exceed the range of "optimal" debris loading in dry coniferous forests recommended by Brown et al. (2003). Brown et al. (2003) considered the tradeoff between the benefits of CWD for wildlife habitat and a source of nutrients for productivity and the potentially negative effects of soil heating and fire hazard. They recommended an optimal level ranging between 11.2 and 44.8 Mg/ha of CWD in warm, dry forests (Brown et al. 2003). A maximum level 50% higher, up to 67.2 Mg/ha, was considered suitable for cool and lower subalpine forest types (Brown et al. 2003).

The postwildfire environment is important not only in terms of debris dynamics, but also for influencing regeneration and future successional trajectories. Postfire environments demonstrate a wide range of conditions even within the southwestern ponderosa pine type, reflecting the complexity of factors and mechanisms that influence vegetation change after burning. Savage and Mast (2005) studied 10 ponderosa pine forests in Arizona and New Mexico that burned severely in the mid-20th century, finding two main patterns of response: (*i*) high pine regeneration that returned the forest to a "hyperdense" condition considered vulnerable to another severe fire (five of 10 sites), or (*ii*) deflection to a nonforested grassland or shrubland state (four of 10 sites). They also found one site with low-density pine regeneration, a response consistent with historical patterns in these forests (one of 10 sites). Given that three to five decades had passed since these fires, Savage and Mast (2005) hypothesized that the alternative states could be indefinitely self-perpetuating rather than successional stages leading back to ponderosa pine forest. Haire and McGarigal (2010) examined spatial landscape patterns on one of the same fires and an additional Arizona fire, finding that the distance to the nearest surviving seed sources was negatively correlated with regeneration for nonsprouting species. They concluded that forest recovery was occurring on these two sites but at slow rates, impeded by large patches without seed sources (Haire and McGarigal 2008). Barton (2002) showed that sprouting oaks became the dominant species after severe fire in a mixed pine–oak forest. Strom and Fulé (2007) extended the same type of comparison using a forest simulation model to project future forest development following the severe 2002 Rodeo-Chediski fire in Arizona. Postfire regeneration was predicted to have a multidecadal to century-long impact on ecosystem composition and structure, with sprouting shrubs forecasted to dominate formerly dense pine sites (Strom and Fulé 2007). The question of whether or not pine forests will recover over various spatial and temporal scales has many implications for management of habitat, natural resources, and carbon dynamics (Dore et al. 2008).

In the present study, our goal is to provide quantitative information about postwildfire woody debris and regeneration dynamics over a broader region than that studied by Passovoy and Fulé (2006). The results will provide a defensible basis for environmental analysis of future fire hazard, as well as information about key forest features such as snags, downed woody debris, and postfire regeneration. We had three specific objectives:

1. to measure snags, fallen trees, surface fuels, fire-surviving trees, and postfire regeneration on a chronosequence of severely burned, unlogged ponderosa pine forests across the state of Arizona;
2. to model nonlinear responses of woody debris and surface fuels as functions of time since fire to provide guidance for assessment of postfire impacts; and
3. to assess postfire regeneration in terms of probable future successional trajectories.

Materials and methods

Study sites

We selected sites within ponderosa pine forests that had experienced severe crown fires (Table 2; Fig. 1), avoiding the area around Flagstaff, Arizona, because it had recently been studied by Passovoy and Fulé (2006) using nearly identical methods. Prior to sampling, we talked with resource managers and consulted fire records to make a list of all the candidate fires. We used the following criteria to select fires: (*i*) fire sites were ponderosa pine dominated (> 67% prefire basal area); (*ii*) fires burned severely (> 67% mortality of prefire basal area); (*iii*) no postfire logging or second fire occurred in the last 30 years; (*iv*) slopes ≤ 45%; and (*v*) minimum contiguous area of severe burning of 260 ha. After field reconnaissance, we selected 11 suitable fires throughout Arizona (Fig. 1). We sampled 14 sites (three of the larger fires had two sites each). Sampling covered all six National Forests in Arizona and Grand Canyon National Park (Table 2). The National Forest sites (other than the Pine Mountain site) had been harvested and grazed, but the National Park site had no tree harvesting and limited grazing prior to 1919. We did not observe evidence of tree harvesting at the Pine Mountain site. All sites were dominated by mature trees prior to fires. The only known postfire conifer planting occurred in some areas of the two Dude fire sites (Jeffrey L. Leonard, Tonto National Forest, personal communication).

Field methods

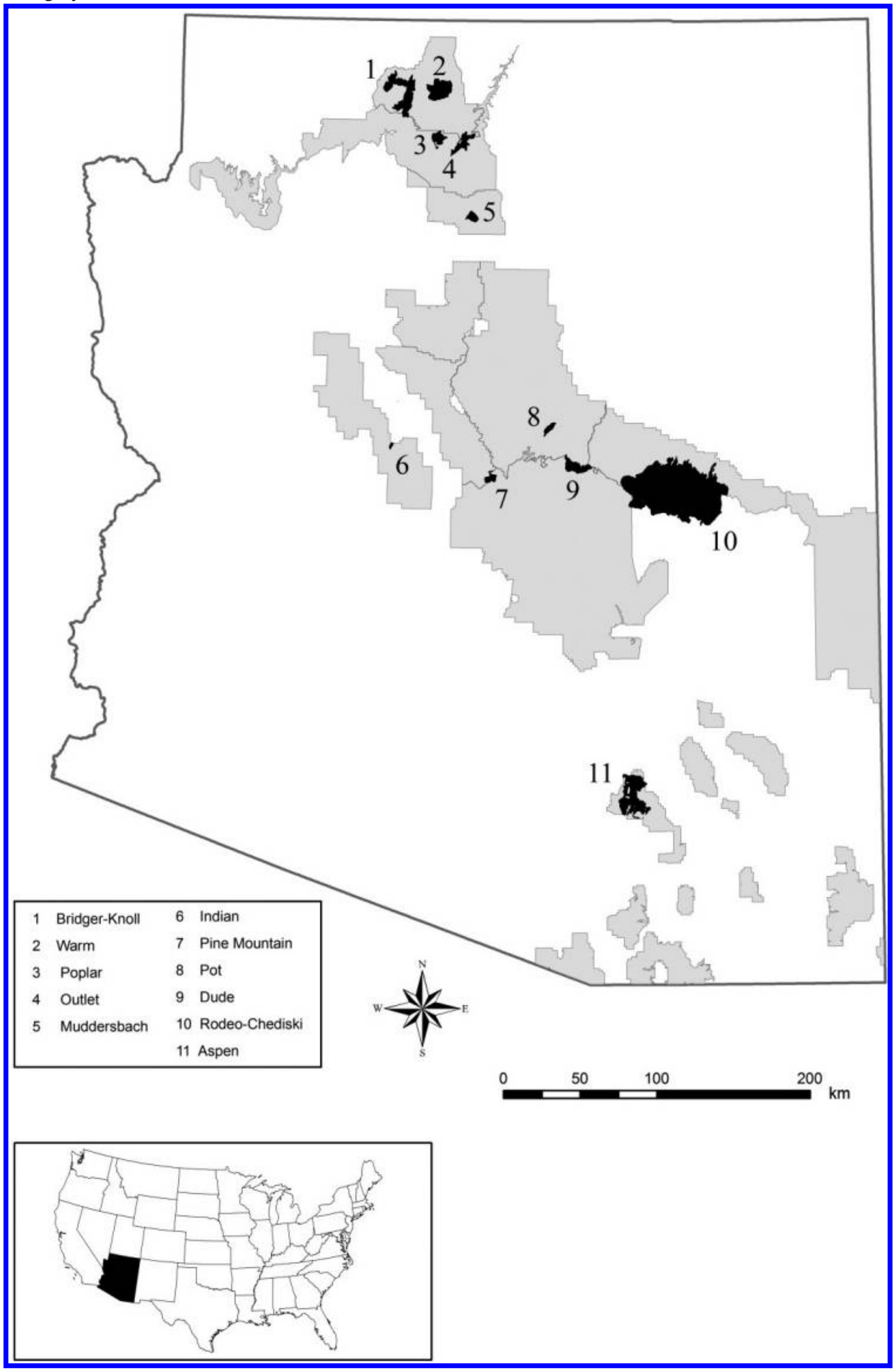
Prior to field data collection, we used a GIS to locate potential plots (spaced 200 m apart) at each site using a random starting point. In the field, we selected plots from among the potential plots that lay within the largest area of severe burning and farthest away from living trees (i.e., unburned forest or trees that survived the fire). We attempted to select plots that were adjacent to each other when possible, but occasionally, adjacent plots were rejected to avoid roads, slopes > 45%, or areas that were not burned severely. We installed thirty 90 m^2 plots at each site except for the Bridger-Knoll

Table 2. Wildfires in Arizona ponderosa pine forests selected for sampling in this study.

Fire	Jurisdiction	Year of fire	Year sampled	Years since fire	Size (ha)	Mean elevation (m)	Postfire overstory species composition
Warm (W)	Kaibab National Forest (North)	2006	2007	1	23 724	2456	PIPO, PSME, ABCO, PIEN, JUOS, PIED, RONE
Muddersbach (Mudd)	Kaibab National Forest (South)	2005	2007	2	2 957	2008	PIPO, PIED, JUOS, QUGA
Poplar (Pop)	Grand Canyon National Park	2003	2007	4	4 711	2576	PIPO, POTR, PSME, ABCO, PIEN
Aspen (Asp)	Coronado National Forest	2003	2008	5	5 591	2266	PIPO, QURU, QUHY, PSME, PIST, RONE, POTR, ABCO, QUGA, QUAR
Rodeo-Chediski (R-C)	Apache-Sitgreaves National Forest	2002	2008	6	186 380	2005	PIPO, JUDE, QUGA, JUOS, ABCO
Outlet (Out)	Grand Canyon National Park	2000	2007	7	5 282	2510	PIPO, POTR, ABCO, PSME, PIEN, RONE
Pine Mt. (Pine)	Prescott National Forest	2001	2008	7	2 827	1648	PIPO, QUGA, QUAR, RONE, JUDE, QUGR
Bridger-Knoll (B-K)	Kaibab National Forest (North)	1996	2007	11	21 600	1993	PIPO, QUGA, RONE, JUOS
Pot (Pot)	Coconino National Forest	1996	2007	11	2 354	2060	PIPO, JUOS
Indian (Ind)	Prescott National Forest	1996	2008	12	535	1796	PIPO, JUDE, QUAR, QUGA, QUGR
Dude (D)	Tonto National Forest	1990	2008	18	8 463	1964	PIPO, QUGR, JUDE, QUCH

Note: Abbreviated fire code is in parentheses; area and elevation were acquired from GIS coverages; species codes are listed in Table 1. All sites were in managed forests except for the Grand Canyon National Park sites (Poplar and Outlet fires).

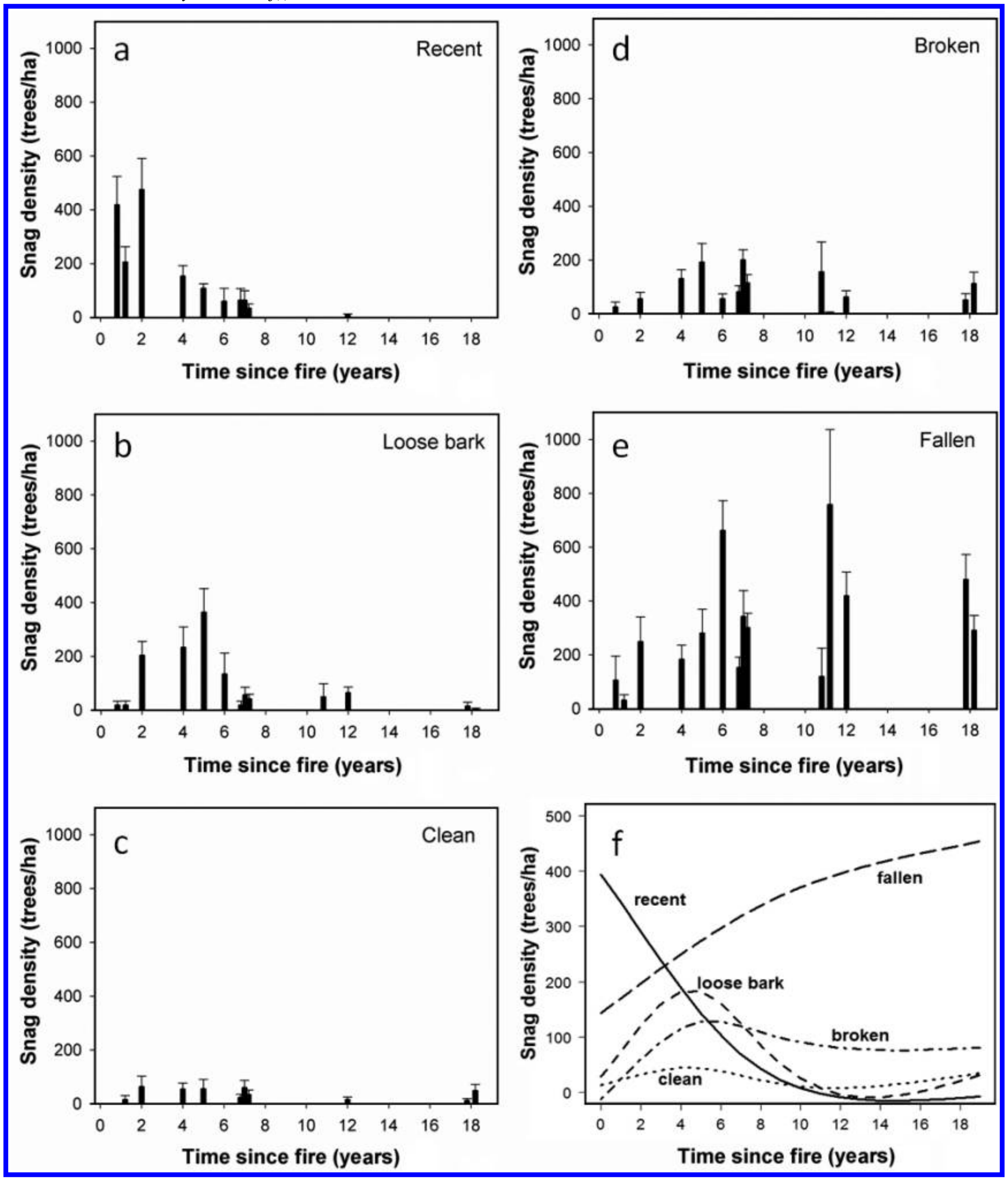
Fig. 1. Locations of the 11 fires sampled for this study in the state of Arizona, USA. National Forest and Grand Canyon National Park boundaries are shown in grey.



site, where we were only able to sample nine plots that fit the selection criteria. Slope, aspect, and elevation were recorded at each plot.

At each plot, we measured forest floor depth, fine woody debris (FWD, < 7.62 cm diameter), and CWD (> 7.62 cm diameter) with a 15 m planar transect (Brown 1974). We inventoried FWD in three size classes (0–0.64 cm, 0.65–2.54 cm, 2.55–7.62 cm diameter) (Brown 1974). These size classes correspond to 1, 10, and 100 h moisture time-lag classes, whereas CWD corresponds to the 1000 h time-lag class. Decay of the CWD was classified following Maser et al.'s (1979) five-class system, where class 1 is a freshly fallen tree with fine twigs present, class 2 is the absence of fine twigs, class 3 is rotten on the outside but solid in the core,

Fig. 2. Relationships between snag densities (trees/ha ± SE) and time since fire for each of five condition classes: (*a*) recent snags, (*b*) loose-bark snags, (*c*) clean snags, (*d*) broken snags, and (*e*) fallen snags. (*f*) GAM-fitted relationships between each snag condition class and time since fire (note different scale on *y* axis in (*f*)).



class 4 is entirely rotten with small, soft, blocky pieces, and class 5 is almost totally decomposed. Classes 1 and 2 were considered "sound" wood and the remaining classes were considered "rotten". We took the 1 and 10 h fuel class measurements in the first 1.8 m of the fuel transect, the 100 h fuel class in the first 3.7 m, and the 1000 h fuel class along the entire length of the transect. We measured duff (fermentation and humus layers) and litter depth (cm) every 3 m for a total of six points per transect. We calculated FWD and CWD biomass using equations from Brown (1974) with southwestern species specific coefficients from Sackett (1980). To capture the complexity of the arrangement of intersecting or "jackstrawed" logs (Passovoy and Fulé 2006), we measured the distance from the intersection of the planar transect with each sampled log to the next log that crossed the sampled log. We also noted the diameter (cm) of every log that touched a tallied log, where "touch" was defined as an intersection of the centerlines of each log.

We measured all living trees and standing and fallen snags on a 90 m^2 (15 m × 6 m) plot centered on each planar transect, thereby linking surface and canopy fuels to the same plot points. We measured species, condition (e.g., live, declining, dead (see snag classes below)), diameter at breast height (DBH, cm), and total height (m) of all living trees and snags

Table 3. Generalized additive model (GAM) results where snag condition class densities and fuel loading or depth are nonlinear functions of time since fire (n = 14).

Response	F ratio	df (est.)	P value	R^2_{adj}	% Dev. Expl.
Snag density by condition classes					
Recent	13.7	2.3	0.001	0.75	79
Loose-bark	2.7	2.8	0.099	0.34	48
Clean	1.6	2.7	0.247	0.18	36
Broken	1.7	2.7	0.231	0.25	41
Fallen	2.5	1.4	0.129	0.16	25
Coarse woody debris loading (CWD)					
Total CWD	6.1	3.5	0.011	0.63	73
Sound CWD	11.8	2.6	0.001	0.74	79
Rotten CWD	3.7	3.6	0.047	0.50	64
Fine woody debris loading (FWD)					
Total FWD	9.5	3.0	0.002	0.71	77
1 h size class	7.1	3.1	0.007	0.64	72
10 h size class	6.4	2.8	0.010	0.60	69
100 h size class	8.7	2.9	0.003	0.69	76
Forest floor depth (FF)					
Total FF	3.4	2.6	0.059	0.41	53
Litter	3.6	2.3	0.053	0.38	49
Duff	6.3	2.9	0.010	0.61	70

Note: df (est.), estimated degrees of freedom used for time since fire; R^2_{adj}, adjusted R^2; % Dev. Expl., the percentage of deviance explained by the model.

taller than breast height (137 cm). We also measured crown base height (m) of living trees. We conducted a visual assessment of each tree (living or dead) to determine if it was alive or dead before the wildfire to estimate tree survival. Regeneration (trees < 137 cm tall) was tallied by species and height class on a 9 m^2 (3 m × 3 m) subplot located within the 90 m^2 overstory plot. For context, Minor (1964) reported an average of 14.3 years (range 12.5–16.2) for ponderosa seedlings to grow to 137 cm in height. State of decay from Thomas et al. (1979) was recorded for all snags: "recent" (snags still retaining bark and fine twigs), "loose-bark" (snags with some bark sloughing off the bole), "clean" (snags with bark mostly fallen off), "broken" (snags broken at some point above breast height), and "fallen" (snags broken below breast height).

Data analysis

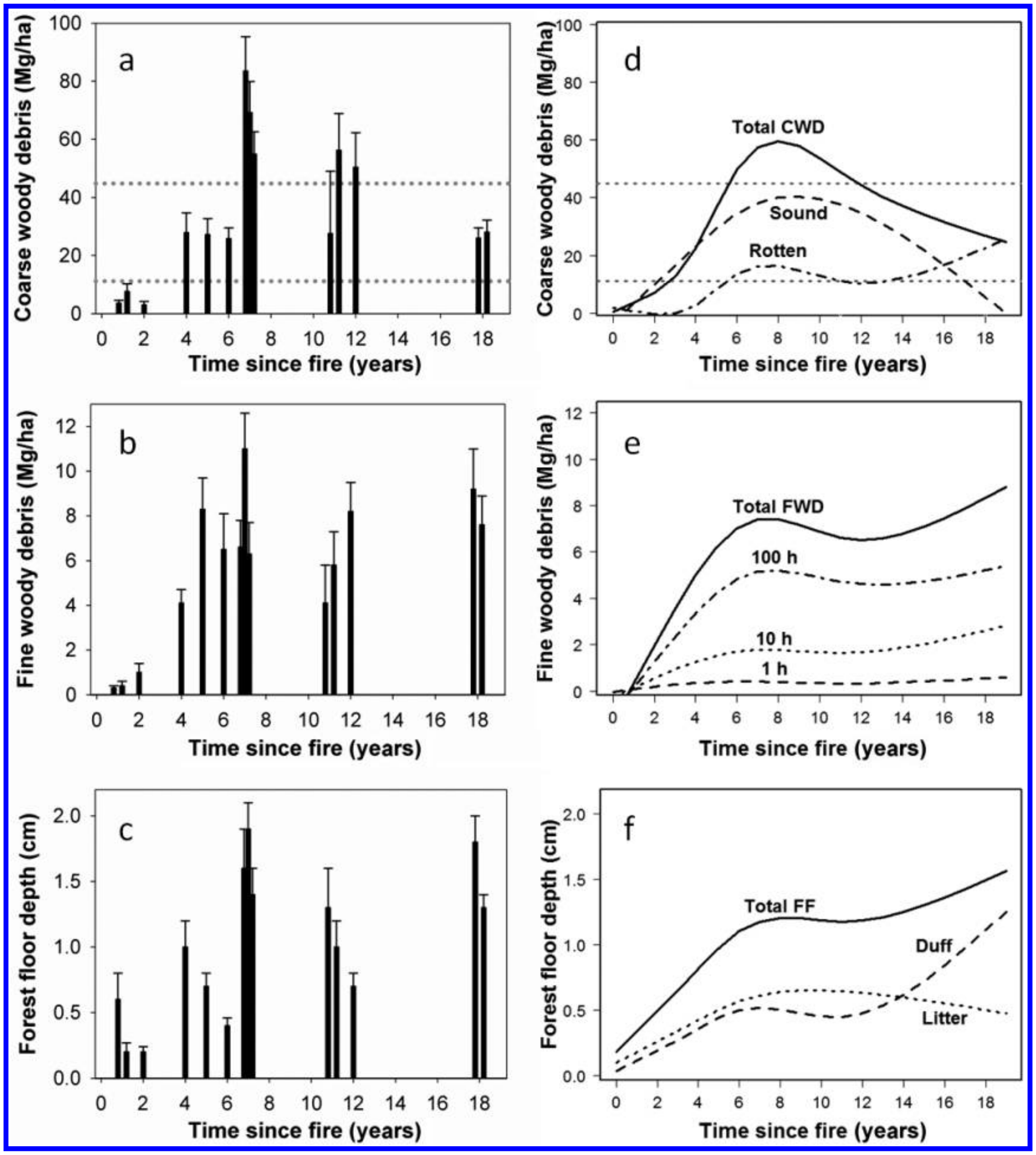
We used generalized additive models (GAMs) to analyze and illustrate the nonlinear relationship between forest structural attributes (i.e., snag density and fuel loading) and time since fire. The GAMs were performed at the site level (n = 14), i.e., average values were calculated across the plots within each site, and these averages were used in the analysis. We used GAMs because previous work has demonstrated that snag density and fuel loading vary nonlinearly over time since fire (Passovoy and Fulé 2006). A GAM is a nonparametric extension of a generalized linear model in which the response is efficiently modeled as a smoothed function of the predictor (Wood 2006). We used the cubic spline basis (bs = "cr") in the "mgcv" package in R version 2.12.0 to fit the models (Wood 2006; R Development Core Team 2010). The nonlinear function is fit by dividing the data into sections. Cubic polynomials are fit to the data in each section with the added constraints that the values and the first and second derivatives are equal at the endpoints of each section. These constraints eliminate any discontinuities in the curve, ensuring a smooth and continuous function along the entire span of the independent variable. The solutions are therefore data-driven and can range from simple linear to complex nonlinear smoothed functions. We limited the complexity of the fit by allowing no more than four degrees of freedom for the time-since-fire term (i.e., we set the parameter k = 5). We show the fitted models to illustrate how average snag densities and fuel loadings at each site vary over time since fire. We chose α = 0.10 to avoid type II errors given the limited sample size of the study (n = 14).

Results

Snag dynamics

Recent snag density was highest in recent fires and then declined rapidly as time since fire increased (Fig. 2*a*). In the fires that occurred 1–2 years ago, mean density of recent snags was 366 snags/ha. Approximately 70% of the stems in the recent fires were classified as recent snags. Mean density of recent snags dropped to 80 snags/ha for the middle-aged fires (4–7 years since fire), and only 1.5 snags/ha remained in the oldest fires (11–18 years since fire). Loose-bark snags were present at all but one site (Pot fire) and were most abundant 5 years following fire (Fig. 2*b*). Clean snags occurred on 10 of 14 sites but had the lowest abundance of any of the snag classes and showed no trend with time; less than 5% of the stems were categorized as clean snags when averaged across all sites (Fig. 2*c*). Broken snags occurred at 13 of 14 sites at relatively low densities, peaking in the midrange of time since fire (Fig. 2*d*). Fallen snag densities rose rapidly in the first few years following fire and remained relatively high through the oldest fire dates (Fig. 2*e*). Time since fire explained significant variation in recent and loose-

Fig. 3. Relationships between fuels and time since fire for (*a*) total coarse woody debris (CWD), (*b*) total fine woody debris (FWD), and (*c*) forest floor (FF). Panels on the right illustrate the GAM-fitted relationships between time since fire and (*d*) total CWD and its components (sound CWD and rotten CWD), (*e*) total FWD and its components (1, 10, and 100 h FWD), and (*f*) total forest floor depth and its components (duff and litter). The horizontal dotted lines in (*a*) and (*d*) represent the lower and upper bounds for "optimal" CWD according to Brown et al. (2003).



bark snags but was not a significant predictor of the other snag condition classes (Table 3). Averaged across all sites, recent, loose-bark, and clean snags > 50 cm DBH (i.e., snags potentially valuable for wildlife) comprised less than 5% of the total snags. However, there was considerable variability from site to site: recent snags > 50 cm DBH ranged from 0% to 24.5%, loose-bark snags > 50 cm DBH ranged from 0% to 9.4%, and clean snags > 50 cm DBH ranged from 0% to 33.4% of total snag density.

Woody debris and forest floor

Coarse woody debris (CWD) loading was much higher at older sites than more recently burned sites, peaking at > 60 Mg/ha in the time period of 6–12 years after fire (Fig. 3*a*). The CWD loadings in the peak time period exceeded the upper limit of "optimal" CWD in dry coniferous forests suggested by Brown et al. (2003), shown with horizontal dotted lines in Fig. 3*a*. Fifty-eight percent of plots in the sites measured 1–2 years after fire had no CWD at all; only 12% of plots had no CWD in the fires older than two years. When CWD was encountered in the 1- to 2-year-old fires, 90%–100% of the wood did not contact any other piece of CWD, but jackstrawed CWD wood contacts did occur in an average of 74% of plots in older fires. Where jackstrawing occurred

Table 4. Mean overstory density (trees/ha) and basal area (m^2/ha) for live trees ≥ 1.37 m in total height at 14 wildfire study sites throughout Arizona.

Fire	Years since fire	PIPO	Other conifer	Deciduous	Total	% survivors
Density (trees/ha)						
W1	1	11.2 (6.2)	0	0	11.2 (6.2)	66.1
W2	1	0	3.7 (3.7)	0	3.7 (3.7)	100
Mudd	2	0	0	0	0	0
Pop	4	0	0	2264.7 (737.3)	2264.7 (737.3)	0
Asp	5	3.9 (3.9)	4.0 (4.0)	1568.3 (666.3)	1576.2 (665.7)	0.5
R-C	6	0	103.8 (46.1)	111.3 (63.5)	215.0 (92.5)	3.4
O1	7	0	0	2645.2 (782.1)	2645.2 (782.1)	0
O2	7	3.7 (3.7)	0	2343.9 (1017.8)	2347.6 (1017.8)	0
Pine	7	252.8 (96.3)	7.6 (7.6)	714.9 (330.0)	975.3 (329.3)	1.9
B-K	11	0	0	504.5 (219.8)	504.5 (219.8)	0
Pot	11	18.6 (9.4)	0	0	18.6 (9.4)	19.9
Ind	12	0	0	92.0 (51.6)	92.0 (51.6)	0
D1	18	148.7 (61.7)	312.0 (164.3)	925.4 (292.0)	1386.1 (304.5)	1.3
D2	18	3.9 (3.9)	105.5 (74.3)	685.5 (226.5)	794.9 (235.6)	1.4
Basal area (m^2/ha)						
W1	1	1.1 (0.8)	0	0	1.1 (0.8)	90.9
W2	1	0	0.6 (0.6)	0	0.6 (0.6)	100
Mudd	2	0	0	0	0	0
Pop	4	0	0	0.1 (0.03)	0.1 (0.03)	0
Asp	5	0.2 (0.2)	0.5 (0.5)	0.3 (0.2)	1.0 (0.5)	70
R-C	6	0	0.4 (0.2)	0.2 (0.1)	0.5 (0.2)	40
O1	7	0	0	0.7 (0.2)	0.7 (0.2)	0
O2	7	0.2 (0.2)	0	0.6 (0.2)	0.8 (0.2)	0
Pine	7	3.7 (1.3)	3.1 (3.1)	1.3 (0.9)	8.2 (3.4)	14.6
B-K	11	0	0	0.4 (0.2)	0.4 (0.2)	0
Pot	11	3.4 (2.4)	0	0	3.4 (2.4)	11.7
Ind	12	0	0	0.1 (0.03)	0.1 (0.03)	0
D1	18	0.2 (0.1)	1.1 (0.7)	1.5 (0.5)	2.8 (0.8)	7.1
D2	18	0.3 (0.3)	0.1 (0.1)	1.3 (0.5	1.7 (0.5)	29.4

Note: Standard errors are shown in parentheses. Fire codes are in Table 2; note that three fires had two study sites each (e.g., W1, W2). PIPO, *Pinus ponderosa*; % survivors, the percentage of the total density and basal area that were alive prior to the wildfire.

frequently enough to calculate an average distance to the nearest CWD contact (fires > 2 years old), the mean distance to the next CWD piece was 2.0 ± 0.3 m and did not show a trend with time since fire. FWD loadings were low in the first few years after fire but rose quickly to a stable level around 7 Mg/ha by 6–12 years after fire (Fig. 3*b*). Forest floor depth had a similar trend with time since fire, rising to approximately 1.5 cm by 7 years after fire (Fig. 3*c*). Time since fire explained significant variation in all the fuel variables (Table 3). CWD switched from predominantly sound to predominantly rotten debris around 16 years after fire (Fig. 3*d*). Forest floor depths were predominantly comprised of duff beginning around 14 years after fire (Fig. 3*f*).

Forest structure and regeneration

Forest structural characteristics and regeneration were highly variable and did not change in a consistent pattern with time since fire. Although a number of the postfire sites were very dense with trees — five of 14 sites had > 1000 trees/ha — only one site had a basal area > 4 m^2/ha (Table 4). Six sites had no surviving trees from before the fire. Of the remainder, 20% or fewer of the trees were survivors except at two recent sites where the survivors were sparse, averaging four–seven trees/ha, but those made up most or all of the trees at these sites (Table 4). Ponderosa pine, the dominant species at all sites prior to fire, was absent from the living tree overstory at half of the sites and represented less than 20 trees/ha at five other sites; only two sites averaged over 100 pines/ha (Table 4). Other conifers were absent at eight sites, and fewer than 10 trees/ha were present at three others; only three sites averaged over 100 other conifers/ha (Table 4). Deciduous species (e.g., quaking aspen, New Mexico locust, several oak species (see Table 1)) were numerically dominant wherever they were encountered, on 10 of the 14 sites. In contrast to conifer density, only one site with deciduous species averaged fewer than 100 stems/ha; nine sites had over 500 stems/ha.

Regeneration (trees < 137 cm tall) was variable across sites (Fig. 4), with standard errors high relative to means (Table 5). Conifer regeneration was very sparse (Table 5). Pines were absent at eight sites and other conifers were absent at 12 sites. Total conifer regeneration exceeded 80 stems/ha at only three sites (two of which had some postfire planting, on the Dude fire), but one fire (Pine Mountain) had exceptionally high pine regeneration, over 11 000 stems/ha. All but two sites had regeneration by deciduous species and they

Fig. 4. Regeneration was variable and included sites with (*a*) prolific aspen regeneration (Outlet fire); (*b*) prolific pine regeneration (Pine Mountain fire); (*c*) abundant oak regeneration (Aspen fire); and (*d*) no tree regeneration (Pot fire).

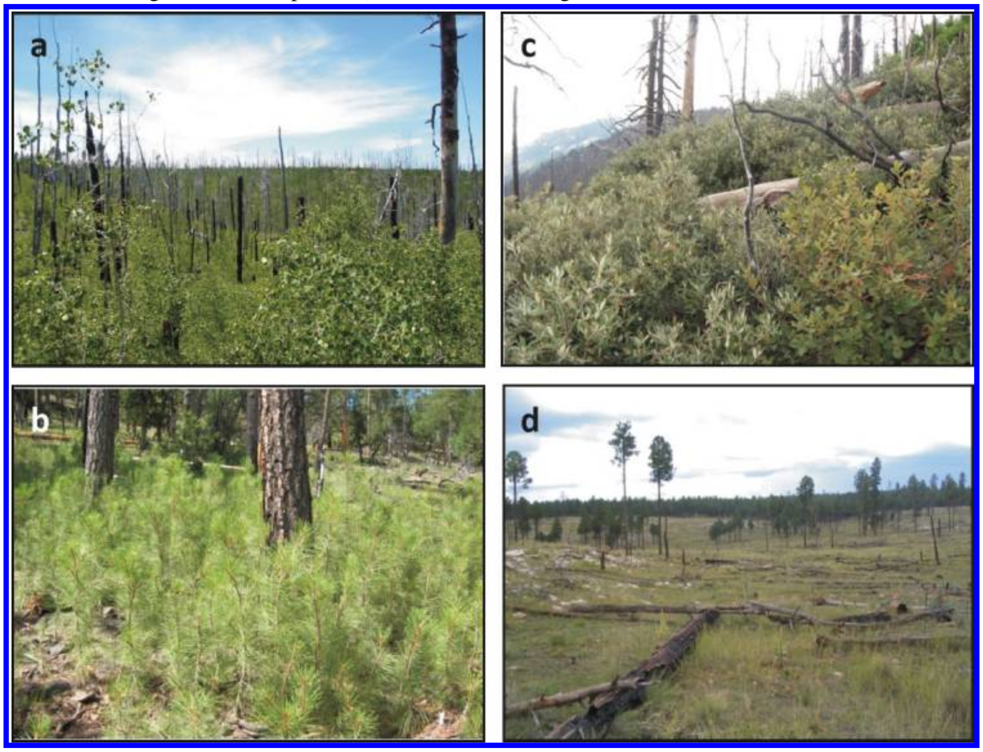


Table 5. Mean regeneration density (stems/ha) for trees < 1.37 m in total height at 14 wildfire study sites throughout Arizona.

Fire	Years since fire	PIPO	Other conifer	Deciduous	Total
W1	1	0	0	373.6 (195.9)	373.6 (195.9)
W2	1	0	0	6762.5 (1769.1)	6762.5 (1769.1)
Mudd	2	37.0 (37.0)	37.0 (37.0)	0	74.1 (51.5)
Pop	4	37.1 (37.1)	0	7485.6 (1812.0)	7522.8 (1822.8)
Asp	5	39.5 (39.5)	0	1987.0 (1020.8)	2026.5 (1019.0)
R-C	6	0	0	853.7 (336.3)	853.7 (336.3)
O1	7	0	37.1 (37.1)	6176.3 (1196.9)	6214.4 (1205.1)
O2	7	0	0	10806.0 (2321.0)	10806.0 (2321.0)
Pine	7	11233.7 (2643.6)	0	3556.3 (1392.1)	14790.1 (2682.2)
B-K	11	0	0	6316.3 (3186.2)	6316.3 (3186.2)
Pot	11	0	0	0	0
Ind	12	0	0	296.3 (231.8)	296.3 (231.8)
D1	18	297.4 (106.2)	0	518.8 (364.4)	816.2 (365.3)
D2	18	336.2 (153.5)	0	112.2 (448.3)	448.3 (183.1)

Note: Standard errors are shown in parentheses. Fire codes are in Table 2; note that three fires had two study sites each (e.g., W1, W2). D1 and D2 had some postfire planting of ponderosa pine. PIPO, *Pinus ponderosa*.

were dominant at all but two of the sites where they occurred (Table 5). Sprouting species dominated: Gambel oak was the majority of regeneration at the Bridger-Knoll fire and aspen sprouted prolifically at the Outlet and Poplar fires. New Mexico locust dominated regeneration at the Warm fire sites. We analyzed potential effects of surface fuels on regeneration but found no significant relationships.

Discussion

Chronosequence studies

The underlying assumption that space can be substituted for time in a chronosequence is always uncertain and may be especially problematic in a study such as this one, which included study sites separated by as much as 400 km, a variety of differences in site conditions and history, and a range from the lower to upper ecotone of ponderosa pine dominated forests. However, this situation is an inherent issue in studies of severe fires because they occur sporadically in space and time, with substantial variation in prefire management history, prefire fuels, and fire intensity. Our choices of burned sites were further constrained by the fact that we sought unlogged sites to measure fuel dynamics, but postfire logging occurred routinely in the US Southwest until the mid-1990s.

Evidence supporting the application of the present data set as a reasonable space-for-time approximation includes the internal consistency of the data, as indicated by the generally significant fit of models with time since fire as a predictor variable that explained a moderate to high amount of variation, as well as consistency with snag, fuel loading, and postfire structure data reported by Chambers and Mast (2005), Passovoy and Fulé (2006), Savage and Mast (2005), and others discussed below. Based on this evidence, we suggest that the present data set is a useful survey of severely burned ponderosa pine sites in Arizona and that resource managers can apply these data and models, with thoughtful caveats, as a guide to estimating changes following future wildfires.

Postfire snag and fuel dynamics

Recent snags varied strongly and significantly with time since fire, but the other snag condition classes were more variable. Model results indicated that the majority of recent snags (≈75%) would be expected to transition to a subsequent category by 7 years after fire (Fig. 2*f*). The relative densities in snag categories closely paralleled the patterns observed by Passovoy and Fulé (2006) in their chronosequence study in northern Arizona, which covered a longer range of times since fire (3–27 years) in a much smaller geographical area. Chambers and Mast (2005) reported that 41% of snags on a severely burned site near Flagstaff fell by 7 years after fire. The rate of loss of recent snags on the fire chronosequence reflects not only snagfall, but also conversion to other snag decay classes. The number of fallen trees increased rapidly in the first years after fire, but the number of loose-bark snags also grew rapidly through the 4- to 5-year period (Fig. 2*f*).

Several of the changes in density of snag categories over time matched the conceptual framework of Thomas et al. (1979), who developed the snag coding system used in this study and widely applied in western US forests. Thomas et al. (1979) postulated that first snags are recently killed, then the standing snags begin to lose fine branches and bark, then bark is gone entirely (“clean” snag), and then the snag begins to break apart. In the present study, the rapid decline in recent snags and the subsequent rise in the loose-bark and broken categories followed the expected patterns, but clean snags had low abundance with no consistent trend with time. A separate analysis at Mt. Trumbull, Arizona, by Waskiewicz et al. (2007) used tree-ring analysis to determine actual death dates of snags representing all the condition classes from recent through broken. They found that the longevity, or time standing since death, of snags was highly variable but the means did follow the time sequence suggested by Thomas et al. (1979). The findings of the present study thus contribute to the evidence supporting the use of snag categories as a rough guide to time since death, which in turn allows managers to make approximations about the range in variability in various snag categories to be expected in the first two decades following severe fire. These estimates may be useful in forecasting future wildlife habitat and carbon storage.

Fuel loads varied strongly and significantly with time since fire, and the models illustrated in Fig. 3 may be useful for management planning. In contrast to the findings of Passovoy and Fulé (2006), this study found the first evidence in the US Southwest of a pattern of CWD loadings exceeding the upper threshold recommended by Brown et al. (2003). An earlier study by Graham et al. (1994) recommended that between 15 and 29.5 Mg/ha of CWD be maintained in ponderosa pine – Arizona fescue stands and that between 11 and 23 Mg/ha be maintained in ponderosa pine – Gambel oak stands in Arizona. Brown et al. (2003) balanced CWD benefits for wildlife habitat and productivity against potentially negative effects of soil heating and fire hazard. They recommended an optimal level ranging between 11.2 and 44.8 Mg/ha of CWD in warm, dry forests (Brown et al. 2003). A maximum level 50% higher, up to 67.2 Mg/ha, was considered suitable for cool and lower subalpine forest types (Brown et al. 2003). We found the peak CWD levels to be around 60 Mg/ha, occurring between 6 and 12 years after fire. This was the same time period as the peak CWD observed by Passovoy and Fulé (2006), except that they found lower levels around 40 Mg/ha. One possible explanation for the difference is that all of Passovoy and Fulé’s (2006) study sites were in second-growth, managed forests, whereas two of the fire sites with the highest CWD in the present study, Outlet 1 and Outlet 2, were located in never-harvested forests of Grand Canyon National Park, which tend to have larger trees (Fulé et al. 2002). The fact that CWD loading exceeded the threshold may be a concern in terms of potential future fire severity and could be considered a justification for postfire logging or other fuel treatments such as prescribed burning. However, this concern may be counterweighed by the fact that such high levels of CWD were not seen at the other sites. CWD loadings in fires measured in subsequent time periods were well below the threshold, both in this study and in that of Passovoy and Fulé (2006).

Future forest trajectories

The burned sites in this study remained very open for as long as 18 years, with only one of 14 sites having a basal area higher than 4 m^2/ha. In contrast, seven of 10 burned sites studied by Savage and Mast (2005) had basal areas higher than 4 m^2/ha. A likely reason for the difference is the greater period of time since fire in the Savage and Mast (2005) study (fire dates 1948–1977), allowing for greater regrowth in the burned areas as suggested by Haire and McGarigal (2010). Savage and Mast (2005) also encountered ponderosa pine regeneration on every burned site. Three of 10 sites had less than 25 pines/ha of regeneration (Savage and Mast 2005), but in the present study, 50% of the sites, and two-thirds of the sites over 5 years since fire, had no pine regeneration at all. There was also no clear trend toward increasing pine regeneration in the older fires.

We found evidence for the alternative stable postfire states suggested by Savage and Mast (2005). (*i*) “Hyperdense” pine regeneration was observed at one site, the 7-year-old Pine Mountain fire. This site also had the highest pine basal area and a relatively high proportion of fire-surviving basal area, suggesting that the presence of seed sources was linked to the high pine regeneration. (*ii*) Evidence of potential deflection to non-pine-forest conditions was observed at eight of the 14 sites, which had no pine regeneration and minimal regeneration by other conifers. In six of those eight sites, the densities of overstory and understory stems of sprouting deciduous species ranged from hundreds to tens of thousands of stems per hectare. These sites showed more abundant regeneration by deciduous species than those studied by Passovoy and Fulé (2006) near Flagstaff, Arizona, but the difference may be related to the unusually high dominance by pine in the latter region with some fire sites naturally having no or few sprouters. Although pines and other conifers may recolonize these sites given enough time, the strong competition from dense, often shrubby, sprouting species could result in a long-term conversion to aspen forest at higher elevations and oak woodland at lower elevations (Barton 2002; Strom and Fulé 2007). The other two potential deflection sites, the Muddersbach and Pot fires, had minimal conifer or deciduous overstory or understory, indicating the potential for conversion to grassland vegetation. Sustained failure of regeneration has been described in the Horseshoe fire in northern Arizona, where eddy-covariance flux tower measurements show that the postfire site remained a net carbon source over a decade after burning (Dore et al. 2008), illustrating the fact that severe fires have consequences for ecosystem services such as carbon sequestration in the long-term. Older landscape scars left by severe fire that caused long-term vegetation type conversion are still observed at various sites around the US Southwest (Haire and McGarigal 2008, 2010) but have not been systematically studied. In contrast to Savage and Mast's (2005) findings, we saw evidence for possible recovery to moderate-density pine forest or savanna at five of the 14 sites, but several also had dense deciduous trees. In sum, compared with the findings of Savage and Mast (2005), the present study found a smaller proportion of "hyperdense" pine sites and somewhat higher proportions of deflected and recovering sites. More time is needed to see whether these patterns will remain stable, but there are substantial obstacles to pine forest recovery: competition with sprouting species and (or) grasses, lack of seed sources, and the forecast of warmer, drier climatic conditions for coming decades (Seager and Vecchi 2010). Where forest does not recover, future fuels and fire behavior will be highly influenced by herbaceous and shrubby plant communities that were not measured in this study.

Management implications of the fuel loadings can be complex. There are limitations in the current fire behavior modeling systems to making full use of CWD loadings in fuel modeling (Passovoy and Fulé 2006), and model outputs have considerable uncertainty (Cruz and Alexander 2010). The thresholds for CWD management developed by Brown et al. (2003) may be only a rough guide, especially as managers seek to balance a variety of competing ecosystem factors such as wildlife habitat and tree regeneration. However, the knowledge of snag and CWD trajectories over time should be helpful for managers to plan ahead for long-term postfire changes. The increasing occurrence of severe fire in the southwestern US and many other areas of the world, associated with warming climate (Seager and Vecchi 2010), means that a rapidly growing fraction of forestlands will have postfire characteristics in the coming century.

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